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Life cycle analysis of energy use and greenhouse gas emissions for road transportation fuels in China

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ABSTRACT

Life cycle analysis is considered to be a valuable tool for decision making towards sustainability. Life cycle energy and environmental impact analysis for conventional transportation fuels and alternatives such as biofuels has become an active domain of research in recent years. The present study attempts to identify the most reliable results to date and possible ranges of life cycle fossil fuel use, petroleum use and greenhouse gas emissions for various road transportation fuels in China through a comprehensive review of recently published life cycle studies and review articles. Fuels reviewed include conventional gasoline, conventional diesel, liquefied petroleum gas, compressed natural gas, wheat-derived ethanol, corn-derived ethanol, cassava-derived ethanol, sugarcane-derived ethanol, rapeseed-derived biodiesel and soybean-derived biodiesel. Recommendations for future work are also discussed.

Contents

1. Introduction

Life cycle analysis aims to assess potential impacts of a product over its full life cycle [1]. It helps to avoid problem shifting between the different processes which are part of the life cycle of a product and has been proven to be a valuable tool for decision making towards sustainability. Life cycle energy and environmental impact analysis for conventional transportation fuels and alternatives such as biofuels has become an active domain of research in recent years [2]. The main life cycle stages for transportation fuels include feedstock recovery and transport, fuel production and transport, and fuel consumption. Feedstock recovery and transport and fuel production and transport together are also known as the "Well-to-Tank" (WtT) stage, while fuel consumption as the "Tank-to-Wheel" (TtW) stage. For instance, the WtT stage of petroleum fuels includes the following main stages: crude oil recovery and transport, fuel refinery and transport. The WtT stage of biofuels includes biomass cultivation (agricultural stage) and transport, biofuel conversion (industrial stage) and transport. Together the WtT and TtW stages are commonly referred to as "Well-to-Wheel", i.e. the whole fuel life cycle. The life cycle framework of transportation fuels is schematically shown in Fig. 1.

The road transport sector is expected to be the dominant oil consumer and a major source of greenhouse gas (GHG) emissions in China considering the rapid growth of private vehicles in recent

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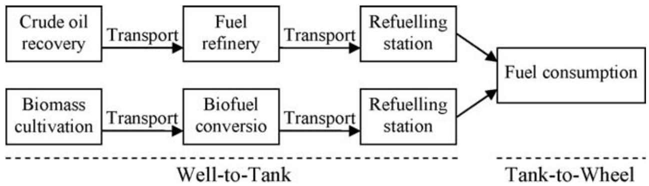

Fig. 1. Life cycle framework of transportation fuels.

years [3]. Several studies have estimated the future trends of fuel demand and GHG emissions in China's road transport sector and/or assessed the effects of various reduction measures such as alternative fuel promotion [4–8]. However, these studies were focused on the TtW stage only, and could therefore be misleading to a certain degree because they did not show the whole picture. For example, it was concluded in [5] that biofuel promotion would contribute largely to the reduction of petroleum demand and only marginally to the reduction of GHG emissions in China's road transport sector. This conclusion was however based on the assumption that the energy stored in biofuel is all renewable bioenergy and thus the use of biofuel would eliminate fossil fuel use entirely, and that a large amount of carbon dioxide (CO_2) would be released during the burning of biofuel just as with fossil fuels. Although this might be correct when only the TtW stage is considered, it is clearly not the case from a life cycle perspective, since biofuel does not exist naturally and the WtT stage of biofuel is usually associated with large amounts of fossil fuel use and much of the CO_2 released during biofuel combustion is actually absorbed from the atmosphere during biomass cultivation and thus should not be counted. It is obvious that the full life cycle should be taken into account for each road fuel when evaluating the effects of reduction measures on future energy demand and GHG emissions in China's road transport sector to avoid problem shifting between different fuel life cycle stages. There is therefore the need to examine the life cycle for road fuels in China.

Conventional gasoline (CG) and conventional diesel (CD) are the two dominant road fuels in China. Liquefied petroleum gas (LPG), compressed natural gas (CNG) and bioethanol (in the form of 10% bioethanol and 90% CG blend) have been introduced in the last decade, while biodiesel is likely to be introduced soon according to the government's renewable energy planning [5]. Fuel ethanol production in China is currently mainly from corn and a small amount from wheat and cassava, with sugarcane as a promising future choice [9]. Options for biodiesel feedstock that could be considered for China include rapeseed and soybean as they are two of the most common biodiesel feedstock under recent scientific investigations [10]. Road fuels to be assessed in the present study will thus include CG, CD, LPG, CNG, wheat-derived ethanol (E-W), corn-derived ethanol (E-Co), cassava-derived ethanol (E-Ca), sugarcane-derived ethanol (E-S), rapeseed-derived biodiesel (BD-R) and soybean-derived biodiesel (BD-S).

When evaluating the life cycle energy use, fossil fuel use is more important than total primary energy use because it is more relevant to the growing concerns over fossil fuel depletion. In addition, petroleum use needs to be assessed since China's road transport oil demand is expected to be a major factor affecting imports, future oil availability and prices globally [5]. The present study attempts to identify the most reliable results to date and possible ranges of life cycle fossil fuel use, petroleum use and GHG emissions for road fuels in China through a comprehensive review of recently published life cycle studies and review articles. Recommendations for future work are also discussed.

2. Methodology

Estimates of WtT fossil fuel use, petroleum use and GHG emissions for road fuels from various life cycle studies are reviewed. Results that can be readily interpreted are presented and compared with each other. Possible reasons for differences between these results are discussed. Life cycle inventory data used in studies specific to China are analyzed in detail whenever available. TtW fossil fuel use, petroleum use and GHG emissions for each fuel in China are then estimated by the authors to arrive at the life cycle results.

In order to compare the results from various studies, fossil fuel use and petroleum use are normalized to MJ fossil fuel consumption and petroleum consumption per MJ final fuel product (MJ/MJ) respectively and GHG emissions are normalized to grams of carbon dioxide equivalent per MJ final fuel product (g CO_2-equiv./MJ). Normalization of results is based on the net energy content of fuels and global warming potentials for different GHG presented in Table 1 whenever these data are unspecified in the original studies. It should be noted that since most studies used lower heating values to represent the net energy content in the final fuel products, results from studies which used higher heating values [37,46,47] are converted to be based on the lower heating value.

3. Well-to-Tank fossil fuel use, petroleum use and GHG emissions

3.1. Fossil fuels

Fossil fuel production systems have been developed for many years and have reached a high technological maturity. Typical estimates of WtT fossil fuel use and GHG emissions for fossil fuels in developed countries are presented in Fig. 2. The variations of these estimates in different countries are mainly due to different assumptions of: technology level (crude oil and natural gas recovery efficiencies, refinery efficiencies, energy efficiencies of various transportation modes, etc.); sources of electricity; distances and modes of transport (related to country territory, energy resource distribution, amount of energy need to be imported, etc.). Different assumptions of fuel specifications also contribute slightly to the variations. Despite these different assumptions, WtT fossil fuel use and GHG emissions in developed countries were estimated to be in the range of 0.15–0.22 MJ/MJ and 10–19 g CO_2-equiv./MJ for CG, 0.09–0.15 MJ/MJ and 7–14 g CO_2-equiv./MJ for CD, 0.14–0.18 MJ/MJ and 6–13 g CO_2-equiv./MJ for LPG, 0.07–0.24 MJ/MJ and 6–13 g CO_2-equiv./MJ for CNG, respectively, and values in European countries and Australia were generally lower than those in North America [11–22]. Reviews in [23,24] yielded similar ranges of results. In addition, a general linear relationship between WtT fossil fuel use and GHG emissions can be observed in Fig. 2. The shares of

Table 1
Net energy content of fuels and global warming potentials of GHG used in the present study.

	MJ/kg	MJ/l
Net energy content (lower heating value)		
CG	44.80	33.152
CD	43.33	37.697
LPG	47.31	25.547
CNG	43.04	5.509
Bioethanol	27.00	21.185
Biodiesel	38.00	33.440
Global warming potentials (100 years)		
1 g of	CH_4	N_2O
equals	23 g CO_2	296 g CO_2

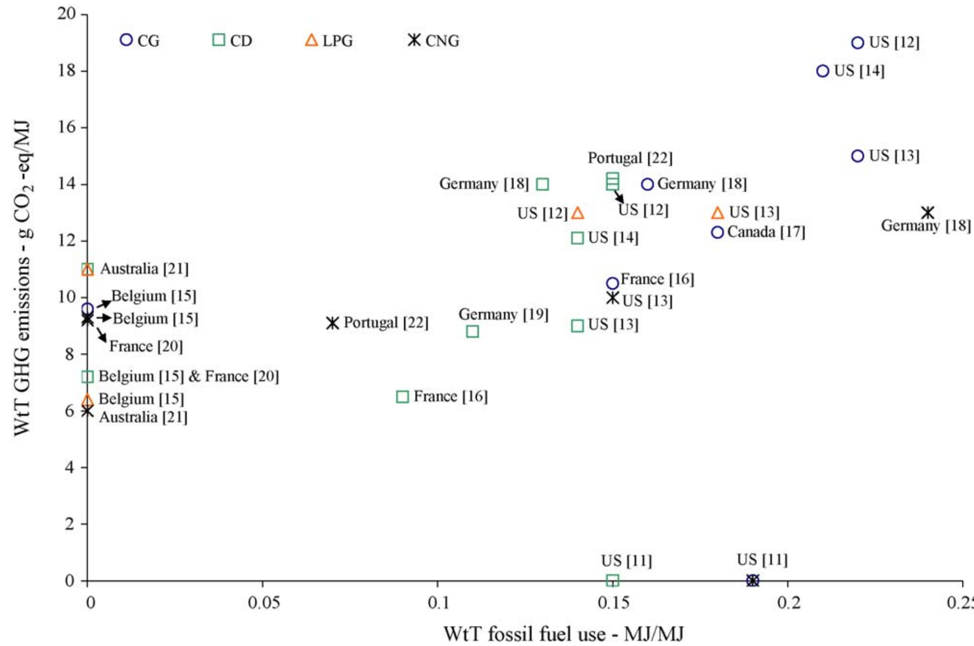

Fig. 2. WtT fossil fuel use and GHG emissions for fossil fuels in developed countries (results from studies which considered only fossil fuel use or GHG emissions are shown on the axis).

petroleum use in WtT fossil fuel use were estimated to be 46–47% for CG and CD [25,26], 44% for LPG [25] and 3–4% for CNG [25,26].

WtT fossil fuel use and GHG emissions estimates for fossil fuel types in China are listed in Table 2. A life cycle model was developed in [27] and used to assess the CG life cycle in China. However, the inventory data were not presented and data sources were unspecified. WtT fossil fuel use estimates in [28] were said to result from the Greenhouse gases, Regulated Emissions and Energy use in Transportation (GREET) model adjusted for China's situation. However, details of the adjustments were not presented. The GREET model was used as a calculation tool to evaluate the WtT energy use and GHG emissions in [29] and their inventory data were considered to be comprehensive and representative for China's situation. For example, data concerning energy use in crude oil and natural gas recovery were on-site collection from 13 oil fields (covering 75% of crude oil production in China in 2004) and 9 gas fields (covering 59% of natural gas production in China in 2004) respectively; data concerning domestic transportation of crude oil, petroleum products, coal and natural gas were from official statistics in 2004; data concerning transportation of imported crude oil were from literature published in 2001; data concerning CNG production were on-site collection from Sichuan province (one of China's major natural gas production areas); energy use and GHG emissions during the refinery of gasoline, diesel and fuel oil were calculated based on nation-wide statistics from the two main oil company Sino-pec and China-petrol; proportions of electricity generation from different sources and energy use during electricity generation from each source were from official statistics in 2004. However, only part of the inventory data rather than a complete inventory data set was presented. Data used in [30] were not presented and data sources were unspecified. Data concerning crude oil and natural gas recovery and transportation used in [31–33] were from the same official statistics published in 2004 and all data were not presented. Life cycle GHG emissions for CD and CNG in China were estimated in [34] using the 'China Version' of a model called 'GHGenius'. The default input data in 'GHGenius' is mainly for North America, while the 'China Version' was created by using specific data and estimates wherever China specific information (not presented however) was available.

Table 2
WtT fossil fuel use and GHG emissions for fossil fuel types in China.

Ref.	Fossil fuel use (MJ/MJ)	GHG emissions (g CO_2-equiv./MJ)
CG		
[27]	0.26	20
[28]	0.24	
[29]	0.27	20
[30]	0.24	22
[31]	0.31	
CD		
[28]	0.19	
[29]	0.2	16
[30]	0.2	
[32]	0.26	16
[33]	0.26	16
[34]		20
LPG		
[31]	0.14	
CNG		
[29]	0.16	16
[30]	0.21	
[31]	0.15	
[34]		12

From the above analysis, results for CG, CD and CNG from [29] were considered to be the most reliable. Results for LPG are less conclusive since only WtT fossil fuel use has been evaluated in [31]. WtT GHG emissions for LPG were taken from [12] since the estimates of WtT fossil fuel use in [31] and [12] are the same. It was also estimated in [29] that the shares of petroleum use in WtT fossil fuel use are 67% for CG (higher than that in [25,26]) and 4% for CNG (consistent with that in [25,26]). These values for CD and LPG in China were assumed to be the same as CG (67%) since they were found to be very close to each other in [25].

3.2. Biofuels

Biofuel production systems are relatively new and more complex compared with fossil fuel systems. Life cycle studies

Table 3
WtT fossil fuel use and GHG emission estimates for bioethanol.

Ref.	Country studied	System boundary[a]	Co-product credits[b]	Fossil fuel use (MJ/MJ)	GHG emissions (g CO_2-equiv./MJ)
Wheat-derived ethanol					
[35]	France	Level 1	No	1.21	
			M	0.52	
			E	0.69	
			MV	0.54	
			R	1.1	
[36]	France	Level 1	*M*	0.58–0.61[c]	43–46[c]
				0.43–0.46[d]	36–38[d]
[16]	France	Level 1	*S&M*	0.49	34
[31]	China	Level 1	–	0.88	
[37]	China	Levels 1 and 2	–	1.01	
Corn-derived ethanol					
[38]	US	Level 1	*S*	0.63–0.83[e]	25–41[e]
			M	0.38–0.46[e]	
[39]	US	Levels 1–3	No	1.29	
			R	1.20	
[40]	US	Levels 1–3	No	0.96	
			M	0.66	
			E	0.58	
			MV	0.83	
			R	0.80	85
[41]	US	Level 1	–	0.74[f]	67[f]
				0.83[g]	113[g]
				0.34[h]	45[h]
[11]	US	–	–	0.61	
[42]	US	Level 1	*R*	0.71	
[43]	US	Level 1	*R*	0.74	57
[44]	China	Level 1	MV	0.93	
[45]	China	–	–	0.91	
[46,47]	China	Level 1	*R*	0.88	94
[31]	China	Level 1	–	0.78	
Cassava-derived ethanol					
[28]	China	Level 1	(MV + R)/2	0.63	
[30]	China	Level 1	–	0.70	65
[48]	China	Level 1	–	0.63	73
[49]	China	Level 1	(MV + R)/2	0.64	75
[50]	Thailand	Level 1	No	0.57	46
Sugarcane-derived ethanol					
[51]	Brazil	–	–	0.12[i]	18[i]
				0.10[j]	17[j]
[52]	Brazil	Levels 1–3	*E*	0.11[k]	20[k]
				0.09[l]	16[l]
[45]	China	–	*E*	0.48	

[a] Level 1 includes direct energy inputs calculated to primary energy level and energy embedded in direct materials inputs, level 2 includes energy embedded in agricultural and industrial machinery, and level 3 includes energy embedded in ethanol/biodiesel plant – represents unspecified (same in Table 4).
[b] No represents no co-product credits, *M* represents mass, *E* represents energy content, MV represents market value, *R* represents replacement, *S* represents system boundary expansion and – represents unspecified (same in Table 4).
[c] It was assumed that the ethanol plant was powered only by natural gas.
[d] It was assumed that half of the energy needed in the ethanol plant was powered by natural gas, the other half was supplied by a straw-fuelled combined heat and power unit.
[e] Higher values correspond to conventional corn cultivation practice while lower values correspond to no-tillage practice in corn cultivation.
[f] Ethanol plants fuelled with natural gas.
[g] Ethanol plants fuelled with coal.
[h] Ethanol plants fuelled with biomass.
[i] Average technology available.
[j] Best technology available.
[k] Current situation.
[l] 2020 scenario.

for biofuels often yield quite diverging results. For comparison purpose, results from various bioethanol studies are shown in Table 3, while those from biodiesel studies are shown in Table 4. Besides various assumptions on inventory data (technology level, climate and geographical conditions, electricity sources and transport), sometimes different assumptions on the following two methodology associated factors also contribute largely to the variations of results:

- *System boundary*: To what extent should the energy use and GHG emissions be included in the system? For example, most studies include only direct energy inputs calculated to primary energy

Table 4
WtT fossil fuel use and GHG emission estimates for biodiesel.

Ref.	Country studied	System boundary	Co-product credits	Fossil fuel use (MJ/MJ)	GHG emissions (g CO_2-equiv./MJ)
Rapeseed-derived biodiesel					
[53]	Sweden	Levels 1–3	No	0.41–0.57[a]	62–88[a]
			E	0.28–0.30[a]	39–40[a]
			M	0.31–0.36[a]	46–51[a]
			S	−0.37–−0.15[a]	31–35[a]
[54]	Lithuania	Levels 1 and 2	E	0.34–0.45[b]	
				0.21–0.28[c]	
[16]	France	Level 1	*S&M*	0.33	20
[18]	Germany	–	–	0.37	25
[19]	Germany	Level 1	*E*	0.38	38
[55]	Sweden	Level 1	–	0.42	
[56]	Various			0.34–0.56	38–53
[57]	Various			0.35–0.65	23–51
[32]	China	Level 1	–	0.28	25
Soybean-derived biodiesel					
[39]	US	Levels 1–3	No	1.32	
			R	1.08	
[40]	US	Levels 1–3	No	0.86	
			M	0.55	
			E	0.3	
			MV	0.55	
			R	0.52	49
[11]	US	Level 1	–	0.38	
[58]	US	Level 1	*M*	0.31	
[59]	Italy	–	–	0.49	19
[32]	China	Level 1	–	0.31	30
[33]	China	Level 1	*M*	0.31	30

[a] The variations were due to different assumptions for the scale of biodiesel plants.
[b] Using common agriculture technologies.
[c] Implementing seed preservation and using biofertilizers.

level and energy embedded in direct materials inputs, some studies [37,54] also include energy embedded in agricultural and/or industrial machinery, while other studies [39,40,52,53] also include energy embedded in biofuel plants.

- *Co-product credits*: Biofuel production systems often generate large quantities of co-products which can be used for other purposes. Therefore, the energy use and GHG emissions should be divided between the main product (biofuel) and various co-products. Commonly used methods to calculate co-product credits include system expansion and allocation. Allocation can be based on physical properties such as mass and energy content, and on market value and replacement value (energy credits equal to the energy required to produce a substitute for the co-products).

E-W was mainly studied for France and China. Effects of different allocation methods were studied for E-W in France [35]. It was found that when co-product credits were disregarded and replacement value allocation was used, WtT fossil fuel use was greater than 1 MJ/MJ. It was much lower when mass, energy content and market value allocation were used. Effects of different power sources for ethanol plants were studied for E-W in France when mass allocation was used [36]. It was found that when the power for ethanol plants were generated from natural gas, the WtT fossil fuel use and GHG emissions would be more than 30% and 20% higher than that if half of the power were generated using straw, respectively. Two co-product credits methods were employed in [16]: system expansion was used during the agricultural stage and mass allocation was used during industrial stage. The estimate of WtT fossil fuel use when mass allocation was used in [35] was in agreement with those in [16,36]. The estimates of WtT fossil fuel use for E-W in China were very high, 0.88 MJ/MJ in [31] and slightly higher than 1 MJ/MJ in [37]. It may appear to be reasonable at first because the energy embedded in agricultural and industrial machinery was included in [37]. However, inventory data in [37] showed that energy embedded in the machinery accounted for less than 4% of the WtT fossil fuel use. The difference of results is probably because co-product credits methods were unspecified both in [31] and [37]. In fact, co-products were not even mentioned in [31], while co-products such as filter cake and vinasse were assumed to be used as fertilizers in [37]. Results from [37] were considered to be more reliable since no inventory data were presented in [31] and a complete inventory data set was presented in [37]. WtT petroleum use and GHG emissions were not studied in [31,37], where future work should thus focus on. The share of petroleum use in the WtT fossil fuel use for E-W in China is estimated by the authors to be 32% based on the inventory data set from [37]. The shares of petroleum in fossil fuel use for agricultural chemical production in the estimation are derived from [28] and listed in Table 5. When comparing the contributions of each stage to the WtT fossil fuel use, significant differences were found between [31] and [37]. As shown in Fig. 3, wheat cultivation (wheat transport was included in this stage in [31]) accounted for 25% of the WtT fossil fuel use, while ethanol production (ethanol transport was included in this stage in [31]) accounted for 75% in [31], similar to estimates in [16,35]. In contrast, wheat cultivation was estimated to account for 73% of the WtT fossil fuel use in [37], where energy embedded in agricultural chemicals (including

Table 5
Shares of petroleum in fossil fuel use for agricultural chemical production (%).

Fertilizers			Herbicides and pesticides
N	P_2O_5	K_2O	
0.5	25.8	29.5	56.7

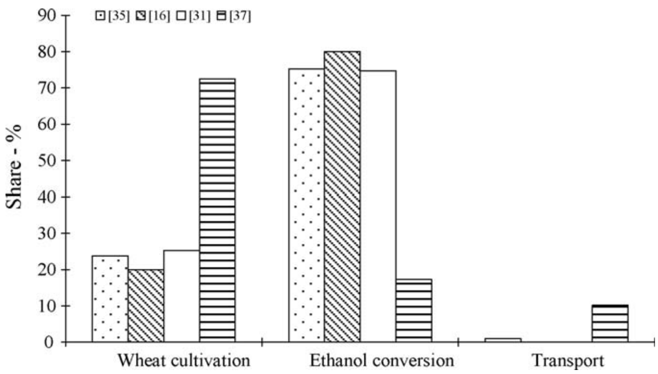

Fig. 3. Contributions of each stage to the WtT fossil fuel use for E-W.

Table 6
Land, materials and energy inputs to produce 1 t of bioethanol from wheat.

	Unit	Ref. [35]	Ref. [37]
Wheat cultivation			
Land	ha	0.47	0.83
N fertilizers	kg	99.2	150
P_2O_5 fertilizers	kg	15.9	62.5
K_2O fertilizers	kg	15.7	50
Fuel for machinery (diesel)	kg	46.9	115
Ethanol conversion			
Wheat	t	3.58	3.59
Natural gas	MJ	18,703	0
Electricity	MJ	5,770	3550

fertilizers, herbicides and insecticides) alone accounted for 47%. It is thus necessary to compare the life cycle inventory data in detail in order to understand the diversion of results. As no inventory data were presented in [31], key inventory data from [37] are compared with those from [35] in Table 6. It can be observed that the amount of fertilizer inputs and agricultural machinery fuel use was much higher in China [37] than that in France [35], not surprisingly because the wheat productivity was considerably lower. Energy requirements during the ethanol production stage were however, significantly and surprisingly lower in China than those in France. This is an indication that energy use in the ethanol production process might be underestimated in [37] and that the WtT fossil fuel use for E-W in China might thus be higher than 1.01 MJ/MJ, although further work is needed to confirm this.

E-Co were mainly studied for the US and China. Ethanol derived from corn via dry milling process is less energy and GHG intensive than that via wet milling and thus more favourable [38,46]. The results for E-Co listed in Table 3 are all based on a dry milling process unless unspecified in the original studies. It is clear that there were large variations in the results for E-Co. WtT fossil fuel use was estimated to be 0.61 MJ/MJ in [11] based on a review of earlier studies. It was estimated to be 0.83 and 0.46 MJ/MJ using the system expansion and mass allocation respectively in [38] and in the range of 0.63–0.83 MJ/MJ in [38,42,43] when replacement value allocation was used. It was also found in [38] that the employment of no-tillage practice in corn cultivation would reduce fertilizer and fuel use and organic carbon release from soil and hence would have lower WtT fossil fuel use and GHG emissions compared with conventional practice. Pimentel and Patzek [39] believed that the WtT energy inputs for E-Co were underestimated in most studies because some of the energy inputs were omitted such as the energy embedded in agricultural and industrial machinery, and ethanol plant. They found the WtT fossil fuel use was greater than 1 MJ/MJ both when co-products were disregarded and when replacement allocation method was used, i.e. the fossil energy required to produce ethanol from corn was larger than the energy stored in ethanol. The main reason for this estimate was believed to be that the input data used were old and unrepresentative of current processes [60]. It was later shown in [40] that even when the system boundary was expanded to include energy embedded in agricultural and industrial machinery and ethanol plant, WtT fossil fuel use was still less than 1 MJ/MJ and energy embedded in machinery and plant in fact only represented less than 2% of the WtT fossil fuel use. It was also found in [40] that when market value allocation was used, the WtT fossil fuel use was the highest, followed by replacement value, mass and energy. Effects of different ethanol plants were analyzed in [41]. It was found that WtT fossil fuel use was higher and WtT GHG emissions were much higher when ethanol plants were fuelled with coal compared with those fuelled with natural gas. The use of biomass instead of coal or natural gas would reduce WtT fossil fuel use and GHG emissions significantly. WtT fossil fuel use for E-Co in China was estimated to be in the range of 0.88–0.93 MJ/MJ in [44–46] and at a lower level of 0.78 MJ/MJ in [31]. It was slightly higher when market value allocation was used [44] compared with replacement value [46], consistent with results in [40]. As no inventory data were presented in [31,45] and more detailed and complete data were presented in [46] than those in [44], results from [46] were considered to be the most reliable. WtT GHG emissions were only studied in [47] and the data used were consistent with those in [46]. They were therefore considered to be the most reliable. The share of petroleum use in WtT fossil fuel use for E-Co in the US was estimated to be in the range of 11–17% [26,41,42]. This value was estimated by the present authors to be 15% in China based on the inventory data for the agricultural stage presented in [46] as no data for the industrial stage were available from [46]. However, this estimation should be fairly reliable because the industrial stage contributes very little to the total petroleum use [42]. When comparing the contributions of each stage to the WtT fossil fuel use, no significant differences were found between various studies. As shown in Fig. 4, ethanol production accounted for 50–69% of the WtT fossil fuel use, while corn cultivation accounted for 30–49% and transport for 0–11%. The contribution of ethanol production in [31] was higher than that in other studies probably because ethanol transport was also included in this stage in [31]. The contribution of corn cultivation was higher in [44] than that in other studies probably because the assumption of lower corn yields per ha (for example, corn yields was assumed to be 6500 kg/ha in [44] while this was 7500 kg/ha in [46]). The contribution of transport in [46] was higher than that in other studies probably because the assumption of the transport distance for corn (300 km) and ethanol (500 km) was higher than that in other studies. Energy embedded in agricultural chemicals represents 13–14% of the WtT fossil fuel use in the US [40,42], while this value is 19% in China [46].

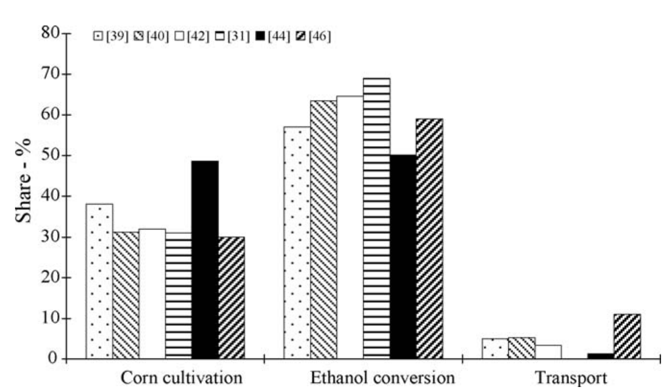

Fig. 4. Contributions of each stage to the WtT fossil fuel use for E-Co.

E-Ca was mainly studied for China and Thailand. WtT fossil fuel use for E-Ca in China was estimated to be in the range of 0.63–0.64 MJ/MJ in [28,48,49] and at a higher level of 0.70 MJ/MJ in [30]. Inventory data were not presented and co-product credits methods were unspecified in [30,48]. Results from [28,49] were the averages of results when market value and replacement value allocation were used. WtT GHG emissions were not studied in [28] and WtT GHG emissions included only CO_2 in [30,48]. Results from [49] are considered to be the most reliable because it is the most recent study with a complete inventory data set presented. The share of petroleum use in WtT fossil fuel use for E-Ca in China was estimated in [28] to be 16%. WtT fossil fuel use and GHG emissions for E-Ca in Thailand were found to be lower than those in China [50], mainly because the share of petroleum use in WtT fossil fuel use was much higher (83%) in Thailand. In all of these studies for E-Ca, co-produced biogas was assumed to be combusted to generate electricity and/or steam for the ethanol plants, which is the main reason why E-Ca has lower WtT fossil fuel use and GHG emissions than E-W and E-Co. The contributions of each stage to the WtT fossil fuel use in different E-Ca studies were compared in Fig. 5. For E-Ca in China, ethanol production accounted for the main part (71–89%) of WtT fossil fuel use, while cassava cultivation accounted for 11–27% and transport for 0–3%. The reasons for the noticeable differences between [48] and [28,49] are unknown because no inventory data were available in [48]. Energy embedded in agricultural chemicals represents 19–25% of the WtT fossil fuel use in China [28,49] and 14% in Thailand [50].

E-S was mainly studied for Brazil. It was found that the WtT fossil fuel use and GHG emissions for E-S in Brazil currently were 0.10–0.12 MJ/MJ and 17–20 g CO_2-equiv./MJ respectively [51,52]. It was expected that those values would be further reduced in the next two decades due to technology improvements [52]. It was found in [52] that cane production and transport accounted for around 90% of the WtT fossil fuel use for E-S in Brazil and ethanol production accounted for the remaining 10%. Energy embedded in machinery and plant accounted for less than 5% of the WtT fossil fuel use [52]. WtT fossil fuel use for E-S in China was cited to be 0.48 MJ/MJ in [45] and no inventory data were presented for examination or comparison. As this is the only estimate found in published literature, it is taken to be the most reliable result at this stage. The much higher WtT fossil fuel use for E-S in China than that in Brazil is probably because of the lower sugarcane productivity [9] and lower energy efficiency in ethanol conversion processes. The share of petroleum use in the WtT fossil fuel use for E-S in Brazil is estimated to be nearly 50% by the present authors based on the inventory data presented in [52] and the petroleum use is mainly attributed to agricultural machine operation and materials transport. This value is assumed to be 30% in China because it is likely to be lower due to higher share of labour input at the agricultural stage and lower share of transport in the whole WtT stage.

BD-R was mainly studied for European countries. The WtT fossil fuel use and GHG emissions were found to be in the range of 0.33–0.42 MJ/MJ and 20–38 g CO_2-equiv./MJ respectively [16,18,19,55]. The effects of different co-product credit methods and different assumptions on scales of biodiesel plants in Sweden were studied in [53]. The WtT fossil fuel use and GHG emissions were found to be highest with no co-products credits, followed by market value, energy and system expansion (note that when using system expansion the WtT fossil energy use was negative, indicating the system was a net energy generator!). Differences in environmental impacts and energy requirements between small-, medium- and large-scale systems for the production of BD-R were found to be almost negligible, mainly because the higher oil extraction efficiency and the more efficient use of machinery and buildings in the large-scale system were, to a certain degree, offset by the longer transport distances. It was also found in [53] that energy embedded in machinery and plant accounted for less than 2% of the WtT fossil fuel use. It was shown in [54] that implementing seed preservation and using biofertilizers could reduce the WtT fossil fuel use significantly (37%) and including rapeseed straw as a co-product would add a further reduction of 33–39%. Reviews in [56,57] yielded a range of 0.34–0.65 MJ/MJ and 23–53 g CO_2-equiv./MJ for WtT fossil fuel use and GHG emissions respectively. The WtT fossil fuel use and GHG emissions for BD-R in China were estimated to be 0.28 MJ/MJ and 25 g CO_2-equiv./MJ in [32] and no inventory data were presented for examination or comparison. As these are the only estimates found in published literature, they are taken to be the most reliable results at this stage, even though they seem to be quite low compared with those in European countries. The share of petroleum use in the WtT fossil fuel use is estimated to be 12% by the present authors based on the inventory data presented in [19] and this is assumed to be the case in China. The contributions of each stage to the WtT fossil fuel use in different BD-R studies seem to differ from each other (Fig. 6), possibly because the data from [54] were before allocation and different co-product credits methods were used from other studies in [16]. For BD-R in China, rapeseed cultivation accounted for the main part (52%) of WtT fossil fuel use, while rapeseed oil extraction and biodiesel conversion accounted for 46% and transport for 2%. Energy embedded in agricultural chemicals accounted for 30–31% of the WtT fossil fuel use [19,55].

BD-S was mainly studied for the US. It was found in [39] that the WtT fossil fuel use for BD-S was greater than 1 MJ/MJ, while this value was much less than 1 MJ/MJ in other studies [11,40,32,33,58,59]. The WtT fossil fuel use was the highest when mass and market value allocation were used, followed by replacement and energy content [40]. Energy embedded in

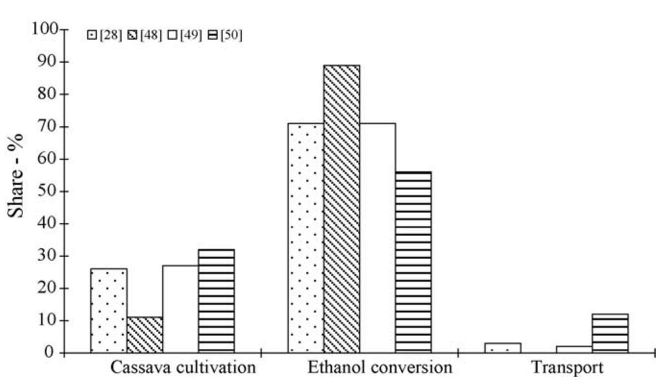

Fig. 5. Contributions of each stage to the WtT fossil fuel use for E-Ca.

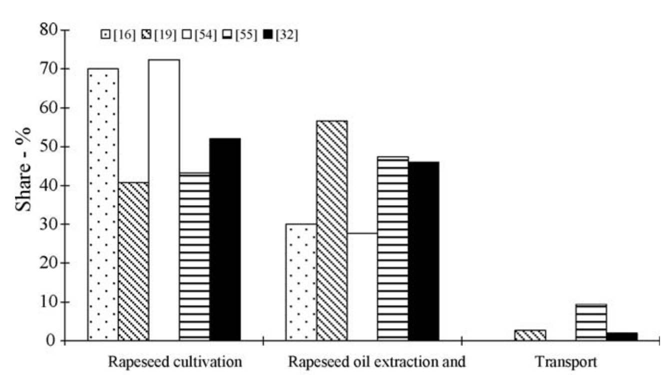

Fig. 6. Contributions of each stage to the WtT fossil fuel use for BD-R.

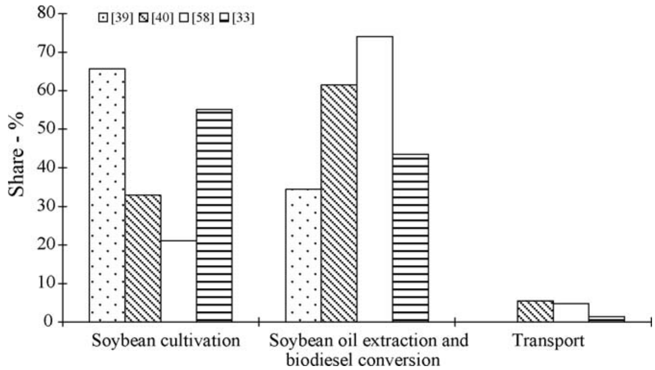

Fig. 7. Contributions of each stage to the WtT fossil fuel use for BD-S.

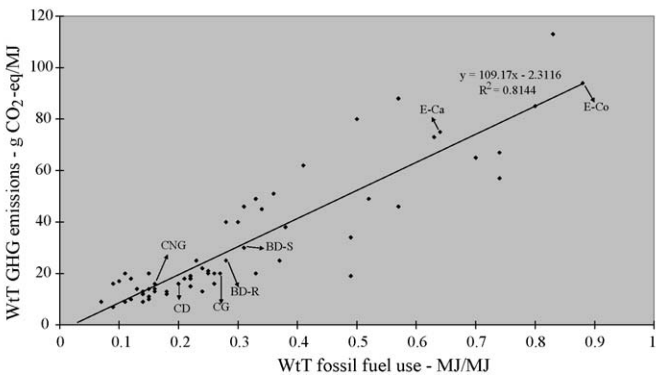

Fig. 8. A linear correlation of the WtT fossil fuel use and GHG emissions for the data points from various studies.

machinery and plant was found to account for 3% of the WtT fossil fuel use [40]. The contributions of each stage to the WtT fossil fuel use in different studies differ from each other (Fig. 7) and the reasons are not clear due to limited data given for examination. For BD-R in China, soybean cultivation accounted for the main part (55%) of WtT fossil fuel use, while soybean oil extraction and biodiesel conversion accounted for 43%. The share of petroleum use in the WtT fossil fuel use is estimated to be 18% by the present authors based on the inventory data presented in [58] and this is assumed to be the case in China.

From the above analysis, WtT GHG emissions for LPG, E-W and E-S in China cannot be determined. However, there should be a roughly linear relation between the WtT fossil fuel use and GHG emissions as shown in Fig. 2. A correlation of WtT fossil fuel use and GHG emissions data for all fuels available in the present review has been derived and is presented in Fig. 8. It can be observed that the most reliable results for CG, CD, CNG, E-Co, E-Ca, BD-R and BD-S in China identified in the above analysis do not show large deviation from the correlation. The WtT GHG emissions for LPG, E-W and E-S in China are thus estimated respectively to be 13, 108 and 50 g CO_2-equiv./MJ using the correlation.

4. Tank-to-Wheel fossil fuel use, petroleum use and GHG emissions

TtW fossil fuel use is 1 MJ/MJ for fossil fuels and 0 MJ/MJ for biofuels. Similarly, TtW petroleum use is 1 MJ/MJ for petroleum fuels and 0 MJ/MJ for non-petroleum fuels. TtW GHG emissions are assumed to include CO_2 only in the present study because other types of GHG were found to contribute little during the TtW stage [61]. TtW GHG emissions are 0 g CO_2-equiv./MJ for biofuels because the CO_2 released during biofuel combustion is absorbed from the atmosphere during biomass cultivation. While TtW GHG emissions for fossil fuels are calculated by assuming all the carbon in fuel will become CO_2 after combustion (shown in Table 7). This method may seem not appropriate because some carbon in fuel will actually be released to the atmosphere in the form of carbon monoxide (CO) and unburnt hydrocarbon (HC). However, it is very difficult to determine the CO and HC emissions during fuel combustion because it depends on many factors such as vehicle engine type and technology, engine operating conditions, emission control technology and age of engine, etc. In addition, this method can better reflect the strengthening emission regulations because CO and HC emissions will be considerably reduced.

Table 7
Carbon content and TtW GHG emissions for fossil fuels used in the present study.

	Carbon content by mass%	TtW GHG emissions (g CO_2-equiv./MJ)
CG	84.6	69
CD	86.5	73
LPG	82.0	64
CNG	75.0	64

5. Life cycle fossil fuel use, petroleum use and GHG emissions

From the present review and analysis, the life cycle fossil fuel use, petroleum use and GHG emissions for each fuel in China can be determined. The results are shown in Figs. 9–11, where the columns represent the most reliable values and the error bars represent the possible ranges.

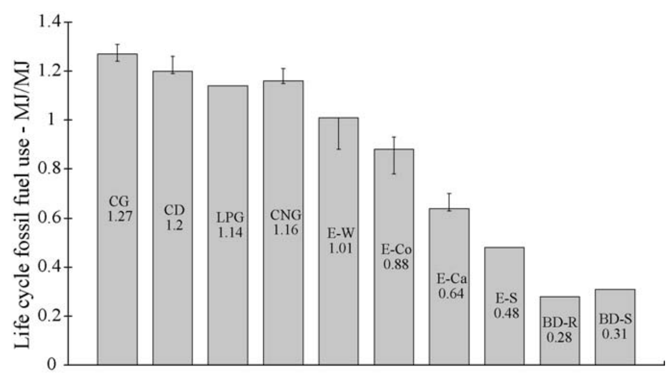

Fig. 9. Life cycle fossil fuel use for each fuel in China.

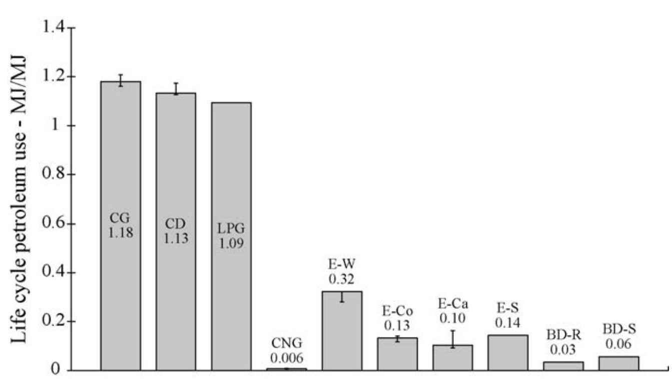

Fig. 10. Life cycle petroleum use for each fuel in China.

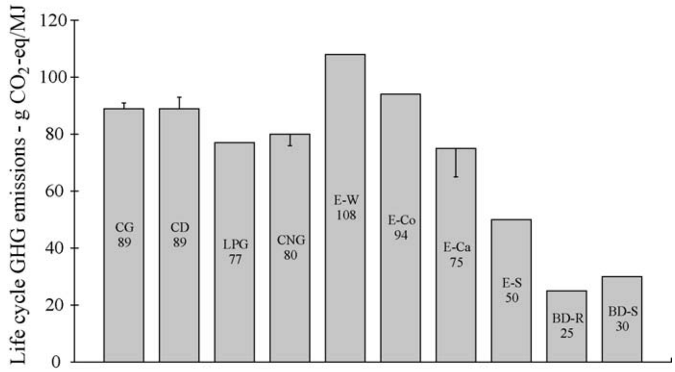

Fig. 11. Life cycle GHG emissions for each fuel in China.

6. Discussion and conclusions

From the present review it can be observed that LPG and CNG have slightly lower life cycle fossil fuel use and GHG emissions than CG and GD in China. Bioethanol from different feedstock varies largely in terms of life cycle fossil fuel use and GHG emissions. E-W and E-Co can offer only moderate fossil fuel use benefits and no GHG emissions benefits at all when compared with CG, while E-Co and E-S seem to be much better choices. Both BD-R and BD-S can offer substantial benefits in terms of life cycle fossil fuel use and GHG emissions. Replacing petroleum-based fuels with CNG or biofuels can reduce life cycle petroleum use significantly. Biodiesel seems to be superior to bioethanol in terms of all three criteria.

During this analysis difficulties arose since inventory data presented in most studies were incomplete and data sources in some studies were not specified. It is therefore recommended that future studies present complete inventory data and specify data sources clearly whenever possible in order to allow examination and revision when more up to date data are available. Because of the data limitations mentioned of the studies reviewed, the results identified in the present study are generally considered to be most reliable for CG, CD, CNG and E-Ca, fairly reliable for E-Co and E-W but not so certain for LPG, BD-R and BD-S, and least certain for E-S. In terms of the different criteria assessed, the results are most reliable for fossil fuel use, followed by GHG emissions and the petroleum use. Whereas life cycle petroleum use was not generally assessed in most studies, it should arguably be given a higher emphasis than fossil fuel use since oil is the most depleted fossil energy resource, particularly with respect to countries having less oil than other energy resources such as is the case in China [3].

While the different co-product credit methods can have a large impact on the WtT energy use and GHG emissions for biofuels, the choice of which method can be rather difficult. As stated in [35], there is no complete justification concerning the reason for choosing one and not a different method and there is no single method which is most appropriate for all biofuel processes. As can be seen in the present review, sometimes using only one method could lead to an underestimation or an overestimation of the WtT energy use and GHG emissions. It is therefore recommended that more than one method should be employed wherever possible whereas only one was used in most studies specific to China.

Energy embedded in machinery and/or biofuel plant has been shown to account for less than 5% of the WtT fossil fuel use [37,40,52,53]. Therefore, excluding these energy inputs is not expected to affect the results significantly. However, it is still worth finding to what degree these energy inputs would affect the results for biofuels in China, since no studies have been found to date providing this information except [37].

Energy embedded in agricultural chemicals usually represents a significant proportion of the WtT fossil fuel use for biofuels as shown in the present review while large differences can be observed between the assumptions employed for this in different studies [28,37,52,55]. It is therefore very important that these data are accurate and extensively specified to yield reliable results. In the Chinese studies reviewed here, these data are however derived either from unspecified sources or from non-Chinese sources such as the GREET model. Energy use and GHG emissions associated with Chinese agricultural chemical production and use needs to be more precisely evaluated to yield more appropriate and reliable results for use in Chinese biofuel assessment.

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